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Priorities in policy and management when existing biodiversity stressors interact with climate-change

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Abstract There are three key drivers of the biodiversity crisis: (1) the well known existing threats to biodiversity such as habitat loss, invasive pest species and resource exploitation; (2) direct effects of climate-change, such as on coastal and high elevation communities and coral reefs; and (3) the interaction between existing threats and climate-change. The third driver is set to accelerate the biodiversity crisis beyond the impacts of the first and second drivers in isolation. In this review we assess these interactions, and suggest the policy and management responses that are needed to minimise their impacts. Renewed management and policy action that address known threats to biodiversity could substantially diminish the impacts of future climate-change. An appropriate response to climate-change will include a reduction of land clearing, increased habitat restoration using indigenous species, a reduction in the number of exotic species transported between continents or between major regions of endemism, and a reduction in the unsustainable use of natural resources. Achieving these measures requires substantial reform of international, national and regional policy, and the development of new or more effective alliances between scientists, government agencies, non-government organisations and land managers. Furthermore, new management practices and policy are needed that consider shifts in the geographic range of species, and that are responsive to new information acquired from improved research and monitoring programs. The interactions of climate-change with existing threats to biodiversity have the potential to drive many species to extinction, but there is much that can be done now to reduce this risk.

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1 Introduction

Biodiversity on earth is in crisis with an unprecedented loss of species (Butchart et al. 2010; Conrad et al. 2006; Laurance 2007; Mooney 2010; Sala et al. 2000; Wake and Vredenburg 2008). Through direct and indirect human activities, species extinction rates are far higher than the background rate of extinction (McCallum 2007; Pimm et al. 2006; Pimm and Raven 2000). There are several main causes of ongoing species extinctions (Lande 1998; Sala et al. 2000; Vitousek et al. 1997), including habitat loss (Pitman et al. 2002), invasive species (Duncan and Blackburn 2004) and resource exploitation (Burgman et al. 2007), although climate change is expected to become more important in coming decades (e.g. Sekercioglu et al. 2008).

Climate-change has now become a major focus of the media (Boykoff 2007) and governments, with dedicated policies and portfolios developed to deal with this major environmental threat (Buhrs 2008; Pyke et al. 2008). Climate-change may have recently surpassed all other environmental causes in terms of public profile (Novacek 2008) with evidence from North America that the majority of people are concerned about climate-change impacts, but are far less concerned about other biodiversity conservation issues (Lindemann-Matthies and Bose 2008; Semenza et al. 2008). Other threats to biodiversity have apparently waned in importance from a public and government perspective (Buhrs 2008; Novacek 2008).

Despite popular opinion, climate-change alone may not be the greatest near-term threat to biodiversity (Lewis 2006; Sala et al. 2000; Secretariat of the Convention on Biological Diversity 2010). In this paper, we argue there are three key drivers of the biodiversity crisis. First are the well understood and globally important threats to biodiversity such as habitat loss and fragmentation, invasive species and resource exploitation. Second, climate-change resulting from increased greenhouse gases in the atmosphere directly threatens some species with extinction, such as corals, coastal specialists and species confined to high elevations (Desantis et al. 2007; Hoegh-Guldberg et al. 2007; Nogue et al. 2009; Parmesan 2006; Wake and Vredenburg 2008). The third driver is the interactions and synergisms between the first two. That is, the combined effects of changing climate and existing threats to biodiversity will multiply the impacts that those processes would have alone, thereby significantly magnifying the biodiversity crisis (Brook et al. 2008; Keith et al. 2008; Sala et al. 2000).

In this paper, our intention is to succinctly review the extent of interaction of climate change with three globally important causes of biodiversity loss: native vegetation loss and fragmentation, invasive species, and resource exploitation. Our first major aim is to enable people working across the broad range of climate-change related fields to understand why these key threats to biodiversity are inextricably also part of the climate-change phenomenon, and therefore why reducing these threats must feature in adaptation measures to climate change. While much effort is already expended countering existing threats to biodiversity, climate-change adaptation now demands new and more efficient approaches, because current efforts in many cases are inadequate.

A major shortcoming of recent climate-change impact studies is that suggested actions do not specifically identify the situations in which a solution may work or who should implement it (Felton et al. 2009; Heller and Zavaleta 2009). The second aim of our paper is therefore to collate well-substantiated and empirically based recommendations from the literature to identify a concise list of the most important actions, policy changes, and players needed to support climate-change adaptation. In doing so, we highlight substantial shortcomings in international, national and regional policy that require urgent attention, in addition to challenges that must be overcome for new scientific approaches to transfer to the policy and decision-making realm.

2 Native vegetation loss and fragmentation interacts with climate-change

Approximately 13 million ha of the world's natural forests are cleared annually (FAO 2005). Land clearing is not only one of the greatest contemporary threats to terrestrial biodiversity, but also one of the greatest threats compounding the impact of climate-change on biota (Millennium Ecosystem Assessment 2005; Sala et al. 2000; Theurillat and Guisan 2001). There are three principal ways that clearing native vegetation may exacerbate climate-change impacts on biodiversity.

First, clearing native vegetation for agriculture or forestry exacerbates climate-change because it is a major source of greenhouse gas emissions world-wide (Gullison et al. 2007). Taking into account reafforestation, Houghton (2003) reported that logging, land clearing and agriculture released 2.2 Pg of carbon per year during the 1990s, which is approximately one third of the amount released by burning fossil fuels (Houghton 2007). Although estimating the amount of carbon released by land clearing remains difficult (Ramankutty et al. 2007), recent figures suggest that forest loss, degradation and loss of peat habitats accounts for 8–20% of anthropogenic greenhouse gas emissions (van der Werf et al. 2009).

A second way that clearing native vegetation strengthens the impacts of climate-change is through its direct influence on regional climates (Deo et al. 2009; McAlpine et al. 2007; McAlpine et al. 2009). Land clearing can increase regional temperature, reduce rainfall and increase weather variability (McAlpine et al. 2007). This increase in extreme weather could compound similar trends in some parts of the globe that are predicted to result from increased atmospheric greenhouse gases (Los et al. 2006; McAlpine et al. 2007).

Third, habitat modification, loss and fragmentation can prevent species from dispersing between remaining habitat patches (Soulé et al. 2004). The resulting reduction and fragmentation of populations interacts with climate-change to magnify the risk of extinction that species face if confronted with just one of these threatening processes (Opdam and Wascher 2004; Travis 2003). A dangerous interaction between fragmentation and climate-change may arise when a fragmented landscape (1) hinders dispersal, preventing species from tracking their climatic niche (Hill et al. 2001; Marini et al. 2009; Primack and Miao 1992), (2) offers a reduced availability of habitat situated in suitable climate space (Huntley 1999; Vos et al. 2008), and (3) harbors small populations which generally possess lower genetic diversity, limiting the potential for adaptation to changing climate (Jump and Penuelas 2005). Dispersal-limited species will be particularly vulnerable to these mechanisms (Thomas et al. 2004; Thuiller et al. 2006).

In summary, given the compounding negative effects of climate-change and habitat loss on biodiversity, there is an opportunity to substantially ameliorate climate-change impacts by conserving and re-establishing native vegetation (Bekessy and Wintle 2008).

2.1 Policies to reduce land clearing

Reducing the area cleared is the most important action to take, because this avoids the extensive difficulties and time needed to effectively restore otherwise degraded habitats (Secretariat of the Convention on Biological Diversity 2010). Actions to address the primary drivers of land clearing should be part of any effective land clearing policy (Lambin et al. 2001). At the most fundamental level, actions to reduce per-capita consumption and population growth are required to reduce demand to clear more land (Ehrlich and Holden 1974), although this is a long-term solution. However, as discussed by Lambin et al. (2001), it is simplistic to link land clearing to per-capita consumption and population growth alone. Land clearing continues at rapid rates in different regions for a range of reasons, which is perhaps why Kishor and Belle

(2004) found very few socio-economic variables significantly associated with land clearing across a dataset spanning 90 countries.

In general, strong governance, such as rule of law, control of corruption, government effectiveness, accountability and political stability (Kaufmann et al. 1999), is an underlying requirement for effective land clearing policy (Gaveau et al. 2009; Kishor and Belle 2004). Within a framework of good governance, a range of national-level financial incentives that promote clearing are important to redress. For example, perverse systems of carbon accounting have provided incentives to clear native vegetation prior to the establishment of carbon sinks or biofuel plantations (Lindenmayer 2009; Pineiro et al. 2009; Schulze et al. 2003), problems that could be quickly eliminated through national regulation, and promoted through international agreements (van Oosterzee et al. 2010).

Land clearing could potentially be decoupled from economic and population growth by improving access to knowledge and technology. Green et al. (2005) observed an approximate doubling in agricultural production from 1960 to 2000 in developed countries despite a slight reduction in the area under production. This was due to improvements in, among others, plant and animal breeding, the use of fertilizer and irrigation. This can have benefits for biodiversity if it reduces the rate of clearing for agricultural development (Ewers et al. 2009; Green et al. 2005). However, the positive effects on biodiversity of focussing agricultural production in a limited area are tempered for several reasons. Intensifying agriculture can be associated with removal of key habitat features at the farm-scale (such as scattered trees, Fischer et al. 2010; Manning et al. 2006). Off-site impacts from agriculture can increase with intensification (e.g. due to run-off of agricultural chemicals), so increasing the efficiency of agricultural inputs must be part of a solution based on intensifying agriculture (Tilman 1999). Further, the intensification of agricultural production is not always offset by land sparing, or a reduction in the net area cleared for agriculture (Ewers et al. 2009). Thus, there must be policy settings that link land sparing with agricultural intensification.

While traditional command and control regulation has a role to play in any effective land clearing policy (Binswanger 1991; Gaveau et al. 2009), mixed success with regulation and enforcement (Borner and Wunder 2008; Kishor and Belle 2004; Tomich et al. 2004) has seen the recognition of other policy approaches to reduce land clearing, such as market-based instruments and financial incentives to protect native vegetation (Ring et al. 2010). Biodiversity offsets is a market-based instrument that has been employed in many countries to reduce the impacts on biodiversity of land clearing (ten Kate et al. 2004). Regulators impose a "cap" on biodiversity loss, and developments can proceed only if any loss of biodiversity can be offset with actions undertaken elsewhere. However, biodiversity offsets deliver no net loss in a narrower range of circumstances than the policy is typically applied (Gibbons and Lindenmayer 2007). Gibbons and Lindenmayer (2007) suggested that offsets will deliver no-net-loss in biodiversity only if: (a) clearing is restricted to highly modified habitats or habitats that will not persist irrespective of pressure to clear, (b) any temporary loss between clearing and the maturation of the offset does not represent a significant risk to biota (see also, Bekessy et al. 2010), (c) gains are sufficient to offset losses, (d) precaution and adaptive management are applied and (e) there is adequate compliance. Thus, biodiversity offsets can only be applied in regions with a governance structure that permits regulation, enforcement and a commitment to no net-loss of biodiversity.

Land clearing also could be effectively reduced at a national or regional level using a carbon trading or taxing system, and, in a kind of global offsets market, using REDD or related mechanisms (Reduced Emissions from Deforestation and forest Degradation, Ebeling and Yasue 2008; Kindermann et al. 2008). REDD is formulated to use market and financial incentives to reduce the emissions of greenhouse gases from deforestation and forest degradation in developing countries. REDD would principally involve monetary payments from developed nations to those developing nations possessing large forest carbon stocks otherwise vulnerable to land clearance. It provides a framework for what is potentially the fastest and least expensive means for reducing global greenhouse gas emissions (Strassburg et al. 2009). However, careful planning is needed to ensure that both reduced emissions and biodiversity conservation goals are met (Corbera et al. 2010; Venter et al. 2009). A modification of REDD, known as REDD+, is an important step towards achieving positive social and biodiversity outcomes in addition to carbon sequestration (Campbell 2009). However, implementing REDD + will require substantial commitment by collaborating governments and non-government agencies to improving data collection, developing appropriate governance and building adequate operational capacity (Burgess et al. 2010). This suite of actions would lead to immediate reductions in the severity and extent of the interaction of climate change with habitat loss.

2.2 Policies to support restoration

Large-scale restoration of native vegetation and the re-establishment of large-scale connectivity is recognised as an essential response to the biodiversity crisis (Gatewood 2003; Jackson and Hobbs 2009; Soulé et al. 2004) and is the most frequently recommended action to counter climate-change impacts on biodiversity (reviewed by Heller and Zavaleta 2009). Effective policy to support the long-term goal of restoration would direct resources for restoration into priority regions, ensure appropriate species are used in restoration programs and link carbon sequestration projects to biodiversity outcomes.

Strategic location of restoration or habitat retention will be an important new approach under climate-change. Restricted-range species are those most likely to be threatened if their movement and dispersal is blocked by habitat loss (Carvalho et al. 2010; Hughes et al. 1996; Steffen et al. 2009; Westoby and Burgman 2006). Therefore, priority areas for conservation action are those likely to be colonised by species with small geographic ranges as a result of climatic shifts. Modelling methods are being developed to help identify priority regions (e.g. Carvalho et al. 2010). Such tools must now be refined and adapted by governments and non-government organisations that are charged with funding, planning and undertaking restoration. The alternative, of haphazardly located restoration, is unlikely to lead to the best conservation investment (Hodgson et al. 2009). Translating prioritization modelling into policy and management will require deliberate effort by scientists, policy makers and managers to bridge the research-policy divide (Anon. 2007).

After regions have been strategically prioritised for restoration, populations of species to be restored must be carefully considered. To avoid introducing new environmental weeds, restoration could use local native species, including individuals from multiple source populations to maximise adaptive potential (Lawler 2009). However, there is concern that shifting climatic niches may render local species less suited for restoration than species from further afield (Hobbs et al. 2009). Introducing species that did not naturally occur in a region in an attempt to pre-empt shifting environmental niches is a risky strategy (Heller and Zavaleta 2009) and there are doubts as to whether it is possible to adequately assess those risks (Ricciardi and Simberloff 2009a, b). Although many translocated species may not become invasive, some intra-continental translocations have had substantial impacts (Mueller and Hellmann 2008), and the enormous impacts of translocations between continents are well known (see section below on Invasive exotic species). Given the poor understanding of the risk that translocated species may become invasive, but the knowledge that invasive species can have very large impacts on biodiversity, assisted migration of non-native species cannot be widely adopted as a routine adaptation measure (Fazey and Fischer 2009; Ricciardi and Simberloff 2009b). National or regional policy and management plans for restoration should reflect this uncertainty.

Nevertheless, there are compelling cases where species are likely to become extinct without ex-situ conservation measures (Rull et al. 2009; Vitt et al. 2010; Williams et al. 2003), and assisted migration may be an important option to consider (Hoegh-Guldberg et al. 2008; Richardson et al. 2009). Hoegh-Guldberg et al. (2008) provided a decision framework to identify cases where translocations are justified. Within that framework, transparent decision-support methods need to be developed and applied (Richardson et al. 2009), requiring close collaboration of scientists and managers. Restoration policy should consider translocations on a species-by-species basis, with translocations justified when there is a high risk of extinction in-situ, when translocation is feasible, and when the benefits outweigh biological and socio-economic costs (Hoegh-Guldberg et al. 2008). Obtaining knowledge about risks and feasibility will require new targeted research (McLachlan et al. 2007).

The third policy response to support habitat restoration makes the link between biodiversity conservation and carbon sequestration. There are great benefits to climate change adaptation of using carbon-sink plantings in a way that enhances biodiversity conservation (Arnalds 2004; Bekessy and Wintle 2008; Lindenmayer 2009; Plantinga and Wu 2003). Policies to subsidize plantings, with the aim to draw carbon from the atmosphere, are now common in many countries (Heath and Joyce 1997; Kula 2010; Zhao and Wen 2010). However, such policies would have greater climate-change adaptation potential if they encouraged biodiversity conservation alongside carbon sequestration (Bekessy and Wintle 2008). Currently, opportunities for conserving biodiversity while storing carbon in vegetation are being lost because national, regional and international policy settings linking carbon and biodiversity are often inappropriate and revisions are urgently needed (Dwyer et al. 2009; Haskett et al. 2010; van Oosterzee et al. 2010).

2.3 Risks of perverse outcomes

With new policy directions, there are new risks of perverse biological outcomes that must be guarded against or removed. What we mean by perverse outcomes are, for example, when native vegetation is cleared to establish carbon sinks, or if exotic species used for revegetation become invasive or alter fire or hydrological processes (Lindenmayer 2009). There are already many examples of revegetation using exotic species which now pose an invasive threat (e.g. Costa et al. 2004; Firth et al. 2006; Harwood et al. 1997; Kotiluoto et al. 2009; Ren et al. 2009), and there are further examples of plans to spread exotic species in an attempt to sequester carbon (e.g. Velez and Del Valle 2007). We suggest that a strong, rapid policy and management response is needed to prevent perverse outcomes of misguided revegetation and carbon sequestration programs.

3 Invasive exotic species and climate-change

Causally related to current global mass extinctions (Pimm et al. 2006) is an accelerating mass invasion event that is several orders of magnitude above prehistoric rates of species range expansion (Ricciardi 2007; Thomas and Ohlemüller 2009). The increasing rate of invasion of non-native species, especially those that traverse continents or move between major areas of endemicity, is a key endangering process for many species (Mack et al. 2000; Pimentel et al. 2005). Invasive exotic vertebrates (Short and Smith 1994), invertebrates (Snyder and Evans 2006), plants (D'Antonio and Vitousek 1992) and diseases (Rachowicz et al. 2005) have taken an enormous toll on native species. Globally, invasive exotic species cost billions of dollars annually to manage, with these costs set to rise as additional species arrive in new regions (McNeely et al. 2001; Xu et al. 2006).

Climate-change is expected to exacerbate problems arising from invasive exotic species (Dukes and Mooney 1999; McNeely et al. 2001; Mooney and Hobbs 2000). Impacts include an increased rate of spread of invasive plants during more frequent extreme weather events (Truscott et al. 2006; Zapiola et al. 2008) and increased competitive ability of invasive plants with increasing CO_2 concentrations (Smith et al. 2000). There is also evidence for increased virulence of pathogens at high latitudes or elevation due to increased temperatures (Laurance 2008; Rahel and Olden 2008; Wake and Vredenburg 2008) and increased sources of invasive species as farming expands into regions previously too cold (Rahel and Olden 2008). There is a risk that mechanisms of introduction will change, particularly through accidental transport and changed patterns of international trade, and there is a risk that current control methods may become less effective (Hellmann et al. 2008). Although not all regions will be equally affected by invasive exotic species (Roura-Pascual et al. 2004), and newly arrived species may provide environmental benefits if they fill vacant niches (Thomas and Ohlemüller 2009), on balance, the interactions of invasive species with climate-change are likely to increase the threat of extinction to native species (Parmesan 2006; Ward and Masters 2007). This is particularly the case with regards to species movements between continents and between centres of endemism (Thomas and Ohlemüller 2009) where there is a strong history of species extinctions (Duncan and Blackburn 2004; Short and Smith 1994).

3.1 Preventing arrival of new invasive species

Given the increased impacts of invasive exotic species that are expected with climate-change, adaptation to climate-change is contingent on better management of invasive species than has been achieved to date. Preventing further introductions is the cheapest and most effective step towards managing invasive exotic species (Keller et al. 2007; Mack et al. 2000; McNeely et al. 2001). Managing the routes by which invasive species enter a new region will be particularly important (Hulme 2009; Hulme et al. 2008). Many invasion routes have substantial industries supporting them. For example, the livestock grazing industry imported and continues to spread exotic grasses in many regions of the world (Bortolussi et al. 2005; D'Antonio and Vitousek 1992; Nichols et al. 2006). Trees introduced for forestry, carbon sequestration or biofuels invade native vegetation and can have substantial impacts on native communities (Becerra and Bustamante 2008; Fine 2002; Pyke et al. 2008; Richardson 1998). The horticulture industry is a major source of invasive plant species and some animals (Goulson 2003; Hingston et al. 2002). For example, in Australia, 70% of invasive weeds are garden escapes and many are still available for sale (Groves et al. 2005). In Mediterranean regions, dry-adapted garden plants posed the greatest risk of becoming invasive (Marco et al. 2010). The pet trade is a fourth major source of invasive exotic species globally (Copp et al. 2007; Lockwood 1999; Rixon et al. 2005). Whittington and Chong (2007) point out that over one billion ornamental fish are traded annually, often resulting in accidental or deliberate introductions and establishment.

The routes of introduction imply that more effort is needed to resolve the conflict between the economic interests of those who import and spread exotic species and human communities who usually bear the cost of invasive species impacts and control (Buckley 2008; Cook and Fraser 2008; Cook et al. 2010). Under the World Trade Organisation (WTO) agreement on Sanitary and Phytosanitary Measures, a country may preclude importation of a new species if there is adequate evidence that it will impact on human, plant or animal life or health. However, this mechanism has limited scope for excluding species or preventing accidental introductions due to contention over what constitutes adequate evidence (Pharo 2006) and because there are political and financial incentives to reduce trade barriers (McNeely et al. 2001). A decade after McNeely et al. (2001) urged resolution of WTO-sponsored invasive alien species, there remains a need for greater international regulation of the risk of spreading invasive alien species through trade (Perrings et al. 2010). Substantial engagement in international negotiations is required to achieve this, including by changing WTO conventions (Cook et al. 2010), and by making better use of links between the WTO and other international conventions, such as the Convention on Biological Diversity (Kahn and Pelgrim 2010; McNeely et al. 2001). Altering these international policy settings would have immediate benefits for conservation in the face of climate change by substantially reducing the rate of arrival of new potentially invasive species.

Besides international agreements, national policy also can have a substantial impact. For example, policies that remove incentives to import raw rather than processed goods could reduce the risk of invasive species arriving accidently in unprocessed materials (Tu et al. 2008). This could involve application of lower tariffs for processed goods compared with raw goods (Tu et al. 2008) (although this may impinge on WTO agreements). Improved quarantine measures are another important step that national governments can take. Effective biosecurity screening is an essential component of climate change adaptation. Current systems are inadequate for identifying potentially invasive species or to prevent accidental introductions. Substantial institutional changes to improve biosecurity are thus needed (Cook et al. 2010; Jefferson et al. 2004; Keller et al. 2007; Mack 1996; McNeely et al. 2001), which in Europe, includes establishment of a new multi-national co-ordinating institution (Hulme et al. 2009). A range of additional national-level approaches to reducing threats from invasive alien species was canvassed by McNeely et al. (2001). Although that work was completed ten years ago, it remains a comprehensive policy guide for adaptation that will quickly reduce the interaction of alien invasive species with climate change.

3.2 Managing established invasive alien species

Management of invasive species that are already established will continue to be essential (McNeely et al. 2001; Panetta 2007; Shah 2001). With an expected delay between the time of arrival and time of becoming invasive (Essl et al. 2011), research is needed to identify species that have already been introduced, but which have not yet become invasive, particularly garden plants (Marco et al. 2010) and pets (Rixon et al. 2005). National or regional policy responses are needed to support risk-reduction measures, including education (Marco et al. 2010). However, policy providing for regulation is essential because competition among sales outlets can increase the likelihood that species known to be invasive will be sold (Peters et al. 2006). Voluntary codes and education alone will not be effective, but in combination with regulation and enforcement, could lead to a rapid reduction in risk.

The development of new technology and adaptation of old technology is proving valuable for limiting impacts of alien invasive species. Fences are widely used to exclude invasive predatory vertebrates, both for short-term protection (Murphy et al. 2003) and for creating long-term "mainland islands" (Moseby et al. 2009; Richards and Short 2003; Saunders and Norton 2001). Such developments have included innovative collaboration of regional government, industry and scientists (Moseby et al. 2009), and non government organisations and scientists (e.g. http://www.australianwildlife.org/AWC-Sanctuaries/Scotia-Sanctuary.aspx). Ongoing development of biological controls (Hoddle 2004), and habitat manipulation (Buckley 2008) may improve the efficiency of current control methods, and enable a broader range of invasive species to be managed. Furthermore, development and adoption of new technologies such as fertility control treatments for feral vertebrates (Jewgenow et al. 2006) or use of population genetics for planning control strategies (Hansen et al. 2007) may substantially improve our ability to ameliorate the impacts of invasive species.

3.3 New invaders and incentives to monitor

Besides greatly improved and expanded efforts to reduce the threat of invasive exotic species, the other critical new approach to managing invasive species under climate-change will be to distinguish between species undergoing range shifts driven by climate change, and species that have been transported beyond their natural capacity to expand. Many species are expected to shift their range with climate-change and new combinations of species may become commonplace (Lindenmayer et al. 2008b). Invasive species management of this class of new species may not be appropriate (Thomas and Ohlemüller 2009) and the decision to eradicate, accept or welcome the new invaders will be case-specific (Walther et al. 2009).

Given the expectation that many species will change their distribution and the uncertainty about the consequences (Schneider and Root 1996), systematic investment in monitoring programs is needed (Likens and Lindenmayer 2011; Lindenmayer and Likens 2010; Lovett et al. 2007; Nichols and Williams 2006). Monitoring will be most useful if it is able to detect range declines in species that are not compensated for by range expansion at a different range margin. Subsequent research may then be targeted to discover why species fail to expand, with likely explanations including barriers to dispersal (Hill et al. 2001), impacts of newly arrived species on declining species (Carroll et al. 2004; McNeely et al. 2001), or impacts of altered competitive or trophic interactions (Tylianakis et al. 2008). Efficient management responses could then be devised, and may include control of an invasive species, habitat restoration or translocations to enable appropriate compensating range expansion (subject to the caveats discussed previously about assisted migration).

Who should do this monitoring? Government-funded, national monitoring programs are already under way in many countries, often in response to obligations under the Convention on Biological Diversity (Pearman et al. 2011; Petit 2009; Reyers and McGeoch 2007). However, a number of approaches are possible for developing effective monitoring programs. Given the extensive role of international conservation organisations in some countries (Milne and Niesten 2009; Schwartzman and Zimmerman 2005), there is the potential for such organisations to lead monitoring projects in collaboration with national or regional governments (e.g. Madoffe et al. 2006). 'Citizen science' projects, funded by governments or NGOs, offer a novel but under-used monitoring approach that has enormous potential for data collection (Devictor et al. 2010; Dickinson et al. 2010). Monitoring by members of the general public could grow substantially by combining smart phone technology (Sutherland et al. 2010) with quality-controlled (such as peer-reviewed) applications and data-bases. This kind of approach is already used for monitoring many aspects of human health (e.g. Gao et al. 2009). Ecologists need to be at the fore-front of the push to design effective monitoring strategies using the full range of innovative tools that are increasingly available.

4 Resource extraction and climate-change

In addition to land clearing, there are several forms of natural resource use that are well recognised threats to biodiversity (Ludwig et al. 1993; Novacek and Cleland 2001). We highlight three examples: water extraction, livestock grazing, and forest logging. We use these examples because they are widespread practices and their impacts on biodiversity are likely to increase by interacting with, or enhancing the effects of, climate-change.

4.1 Water extraction

Extraction of fresh-water, including river regulation, has substantially altered natural water flows, impacting on freshwater biodiversity (Cumberlidge et al. 2009; Dudgeon et al. 2006; Kingsford 2000). Reduced and altered flows of freshwater are threatening processes in estuarine systems (Lamberth et al. 2008; Whitfield 2004), floodplains (Dunham 1994; Kingsford 2000), rivers (Taylor et al. 2008; Walker and Thoms 1993), streams (McKay and King 2006), ephemeral water bodies (Smit and Vanderhammen 1992), mound springs (Ponder et al. 1995), and ground-water ecosystems (Hancock 2002). Climate-change is likely to cause reduced precipitation and runoff in many regions of the world (IPCC 2007) and therefore will act in concert with water extraction. A number of mechanisms could be exacerbated, including increased risk of disease with lower water flow (Johnson et al. 2009), increased concentration of pollutants (Nieuwoudt 2008), altered water temperatures (Matulla et al. 2007), misalignment of the reproductive cycles of aquatic organisms with changing water flows (Gehrke et al. 1995), the creation of barriers to dispersal (Benstead et al. 1999), and more generally, the complete loss of wetland habitat (Deacon et al. 2007).

Ameliorating the combined impacts of reduced rainfall and water extraction would involve allocating more water to environmental flows (Dudgeon et al. 2006; Hancock 2002). Achieving this will require a range of policy changes, including better use of international agreements on climate, biodiversity and desertification (Duda and El-Ashry 2000). Better regulation of water extraction is needed, including policy and policing (Ghosh and Ponniah 2008). Reducing demand for water extraction is essential (Deacon et al. 2007), and may be achieved with a diverse range of approaches such as using crops that require less water (Naylor et al. 2007), desalination or recycling (Dolnicar and Schafer 2009), appropriate price signals and restrictions (Kenney et al. 2008), public education (Syme et al. 2000), and by reducing human population growth (le Blanc and Perez 2008). Delivering environmental flows will have immediate effects for some species (Kingsford and Auld 2005), although recovery of long-lived forest ecosystems may take decades or longer (Hughes and Rood 2003).

4.2 Livestock grazing

A reduction in rainfall in some regions (IPCC 2007), leading to increased drought, is also likely to increase the impact of grazing on some native species. Grazing livestock in uncleared rangelands and other types of remnant native vegetation can reduce biodiversity by direct consumption of palatable species (Landsberg et al. 2002), by altering the vegetation structure (Martin and Possingham 2005), by removing key food resources for native herbivores (Woinarski et al. 2005), and by altering soil properties (Yates et al. 2000). In dry years, when resources are generally limited, the impacts of grazing can be substantially larger because plants are more completely removed (Yarnell et al. 2007), leading to increased erosion and soil dryness (Ureta and Martorell 2009). Furthermore, grazing impacts may be highest in areas with low productivity (Lunt et al. 2007; Milchunas et al. 1988; Proulx and Mazumder 1998). In regions where rainfall declines with climate-change, productivity may be reduced, leading to increased grazing impacts. Finally, livestock impacts may be higher during drought if grazing occurs in areas that are usually set aside for conservation (Lupis et al. 2006; Morton 1990; Retzer et al. 2006). The additional stress of grazing in these refuges may appreciably increase the risk of extinction of some native species (Frank and McNaughton 1992; Retzer et al. 2006).

Solutions to these emerging issues include protecting conservation areas from "emergency" grazing. This would be achievable if conservative stocking rates were used rather than an opportunistic rate, the former being potentially more economically rewarding in addition to reducing pressure on biodiversity during drought (Campbell et al. 2000; Thurow and Taylor 1999). Commercial destocking during drought may be an option in some cases, such as the 2006 drought in Ethiopia (Abebe et al. 2008), however such international solutions would promote opportunistic stocking rates with associated environmental risks. New knowledge of ecosystem services provided by biodiversity could be an additional approach to motivate species conservation on grazing lands (Jackson et al. 2007). Delivering that motivation will require more support for ecological research combined with improved avenues for communication and outreach (Jackson et al. 2007). The Diversitas program (http://www.diversitas-international.org/) promotes such a strategy at an international level, although policies that support conservative stocking rates, ecosystem service research, and outreach are within the realms of all levels of government, such as through financial incentive schemes (Hacker et al. 2010; Rissman 2010) or by providing off-farm income options to ease financial pressures on farms (Easdale and Rosso 2010).

4.3 Forest logging

Our third example highlights ways that forest logging is likely to interact with the effects of climate-change to reduce biodiversity. Climate-change will cause large disturbance events to become more frequent (Allen et al. 2010; Cary 2002; Lenihan et al. 2003; Williams et al. 2001), widespread (Flannigan et al. 2005; Government of British Columbia 2009), intense (Emanuel 2005), or all of these (Franklin et al. 1991; Lenihan et al. 2003; Lewis et al. 2011). This increase in disturbance will magnify the threat to biodiversity posed by exploitative forestry operations. Of particular concern is the potential for more widespread post-disturbance (salvage) logging as the area of disturbed forest increases (Lindenmayer et al. 2008a; Spittlehouse and Stewart 2003). Post-disturbance logging reduces biodiversity through the loss of mature and dead trees, (Franklin and Agee 2003; Lindenmayer and Noss 2006), mechanical disturbance (Jonasova and Prach 2008), and the establishment of exotic plantation trees (Crisafulli et al. 2005; Sessions et al. 2004). Logging prior to disturbance will also have a negative impact by removing resources that are critical to survival of many species in the post-disturbance environment (Mazurek and Zielinski 2004; Pharo and Lindenmayer 2009). Sustainable forest management may therefore require an increasing amount of retained, unlogged elements as climate change becomes more severe. An additional interaction between logging and climate-change may increase the frequency and scale of fires in wet forest regions, to the detriment of biodiversity (Allen et al. 2010;

Cochrane and Barber 2009; Lindenmayer et al. 1999; Thompson et al. 2007). Logging can increase the risk of ignition and provide fuel conditions that would support intense fire (Cochrane and Barber 2009; Lindenmayer et al. 2009; Thompson et al. 2007), while climate-change is likely to increase the occurrence of dangerous fire weather (Flannigan et al. 2009; Williams et al. 2001). Reducing this risk will require reduced forest exploitation (Cochrane and Barber 2009; Lindenmayer et al. 2009; Thompson et al. 2007). Careful regulation of post-disturbance logging also will be needed, including reservation of key parts of landscapes (e.g. biodiversity hotspots and riparian areas), a reduction in post-disturbance logging intensity, and consistent use of indigenous species in reafforestation programs (Lindenmayer et al. 2008a).

We acknowledge that there are ecosystems facing similar dilemmas in addition to the three we have highlighted, notably exploitation of coral reef or coastal communities that are subject to hurricane damage (Hughes et al. 2003; Michener et al. 1997; Nystrom et al. 2000), and freshwater and floodplain environments that are downstream from mines in regions that will be subject to more extreme rainfall events (Lin et al. 2006; Swales et al. 1998). We suggest that re-evaluation and modification of the way that natural resources are managed is a critical form of adaptation to climate change because many impacts that might have been manageable previously are likely to become substantially more difficult as climate-change interacts with commonplace use of natural resources.

5 Conclusion

In the face of global climate change, many ecological systems will adapt, transform or disappear, with the outcome eventually dictated by the success of climate change mitigation efforts. Policy makers at regional, national and international levels, land managers, and conservationists have the task of trying to maintain biodiversity and functioning ecosystems in a world of climate change. This requires a halt to the biodiversity crisis (Butchart et al. 2010; Mooney 2010) by undermining the main driving processes. We argue there are now three key and emerging drivers of the biodiversity crisis: (1) well known existing threats to biodiversity, including habitat loss and fragmentation, invasive pest species and resource exploitation, (2) direct effects of climate change, driven by increasing greenhouse gases in the atmosphere and (3) the interactions and synergisms between existing threats and climate-change. Addressing this reality requires that we do not see climate change mitigation and biodiversity preservation as an either/or trade-off, nor uncertainty as a reason for delaying action (McLachlan et al. 2007). Climate-change adaptation is intrinsically linked to reducing threats to biodiversity.

An essential global response to the worsening biodiversity crisis is to address the fundamental drivers of global change. These are increasing human population, increasing rates of resource consumption, and increasing greenhouse gas emissions (Ayres 2000; Cohen 1995; McMichael et al. 2003; Pimentel 1994; Pyšek et al. 2010). However, there is a range of critical regional responses that policy makers and land managers can take now that will mitigate some of the worst impacts of climate-change on biodiversity (Steffen et al. 2009). Increased effort to combat existing threats will substantially diminish the third driver of the biodiversity crisis. The current efforts to combat existing threats are inadequate and much greater effort is needed using existing and new approaches.

To help guide these efforts we have summarized critical policy and management actions that represent the front line of a thorough climate-change adaptation response (Table 1). Three key trends emerge. First, there are many policy and management actions that can be

Table 1 A summary of key climate change adaptation responses that are likely to result in biodiversity benefits. Within each broad goal we recommend a number of actions and identify who the acting agency may be. The fourth column (Time) lists the approximate potential timeframe within which resultant biodiversity benefits would begin to accrue once the action has been enacted: fast: immediate to a few years; medium: a few years to a few decades; slow: decades to centuries

Broad goal	Recommended action	Acting Agency	Time
Reduce land clearing	Address fundamental drivers of land clearing: per-capita con sumption and population growth;	International/ national/regional policy	slow
	Support strong governance	International/ national/regional policy	medium
	Remove taxation and other financial incentives to clear land	National/regional policy	fast
	Eliminate perverse carbon-accounting rules that promote forest clearance for plantations	International/ national/regional policy	fast
	Improve access to technology enabling higher agricultural production in a smaller area[a].	International/ national/regional policy	medium
	International carbon market and REDD+	Bilateral national agreements, international conventions	fast
	Application of regional market-based instruments, including offsets for no net biodiversity loss.	National/regional policy	fast
Restore native vegetation to create carbon sinks and to enhance biodiversity	Identify priority areas for restoration	Scientist/policy maker collaboration	slow[b]
	Create guidelines to choose species for restoration	Scientist/policy maker collaboration	slow[b]
	Use carbon pricing schemes to fund revegetation	International/ national/regional policy, local implementation	slow[b]
	Guard against perverse outcomes, especially invasive species risk in new plantings.	National/regional policy and enforcement	fast
Prevent new introductions of potentially invasive species	Urgent modification to WTO international agreements	International policy	fast
	Remove incentives for imports with high risk of accidental introductions	International and national policy	fast
	Improve quarantine and other national policies	National/regional policy	fast
Manage established alien invasive species	Identify potentially invasive species before they escape captivity and implement policy to support education and regulation	Scientist/ government collaboration, National/regional policy	fast
	Create new policies to regulate sales of potentially invasive garden plants	National/regional policy	fast
	Apply new and existing technology	Scientist/NGO/ government collaboration, National/regional policy	fast - medium

Table 1 (continued)

Broad goal	Recommended action	Acting Agency	Time
Detect range changes of concern	Establish targeted monitoring programs using both scientists and citizens as primary data collectors	Scientist/NGO/ government collaboration, International conventions, National/regional policy	medium - long
Alter management of natural resources including:			
Water management	Ensure environmental flows are adequate	Scientist/government collaboration, National/regional policy	fast - medium
	Provide international framework to support water management for biodiversity	International conventions, National/regional policy	fast - medium
	Regulate and police water extraction	National/regional policy	fast - medium
	Reduce demand	National/regional policy	fast - medium
Livestock grazing	Implement conservative not opportunistic stocking rates	National/regional policy, land managers	fast - medium
	Exclude stock from land set aside for conservation, especially during drought	National/regional policy, land managers	fast
	Use outreach programs to educate land owners about ecosystem services	Scientist/policy maker/land manager collaboration	medium
	Financial incentive schemes for stewardship	National/regional policy, land managers	fast
	Off-farm income support	National/regional policy	fast - medium
Forest logging	Retain habitat features to provide native species with resilience to disturbance	Scientist/policy maker collaboration (to identify features needed), National/regional policy, land managers	fast - slow[c]
	Reduce and regulate post-disturbance logging	National/regional policy, land managers	fast
	Use indigenous species in reafforestation	National/regional policy, land managers	fast - slow[d]

[a] This approach has risks of perverse outcomes because it does not always lead to increased land sparing. The conditions under which this approach may be beneficial must be carefully assessed

[b] Although changes that bring about revegetation may be fast, the benefits to biodiversity of revegetation are likely to be slow to accrue because forest communities will take decades or longer to re-establish

[c] Species specific; invertebrates may use retained habitat immediately, old-growth specialists may use retained habitat only after the surrounding logged forest has substantially matured

[d] Fast for avoided introduction of invasive species, slow for the time it takes for revegetation to establish

taken now and would result in a rapid reduction in the threats to biodiversity. These are principally actions that circumvent further impacts such as avoiding the introduction of new invasive species and preventing further habitat loss or degradation (Table 1). Nevertheless, actions that pay off in the medium and long-term remain essential for an effective program of adaptation to climate change. A second trend highlighted in our review is the importance of international agreements in driving or resolving threats to biodiversity (Table 1). Climate change adaptation is intrinsically linked, not just to international climate change conventions, but also international trade and conservation conventions. National effort to combat the effects of climate change must include engagement in such international negotiations, particularly those associated with international trade. The third trend emerging from our review is the importance of developing new collaborations between government, NGOs, industry, land managers and scientists to ensure better knowledge transfer, better policies and better on-ground delivery of programs. We have identified specific areas where particular groups must work together to transfer knowledge into practice via policy (Table 1).

An unfortunate by-product of the complex interaction between climate change and biodiversity loss, is the potential that key responses will be delayed. This is based on the assumption that many impacts and outcomes are uncertain and greater efficiencies will be achieved as our understanding improves. It is much easier to delay decisions under the justification of "inadequate information" than to embark on the difficult processes of informed decision making (Nichols and Williams 2006). We have shown, however, that for the vast majority of major threatening processes to biodiversity, sufficient ecological knowledge and policy options currently exist for effective adaptation efforts to be implemented or improved upon, today (Hunter et al. 2010). Policy makers and land managers can take practical action now to reduce the impacts of climate change on biodiversity (Table 1). Such actions will critically determine the trajectory that the biodiversity crisis will take over coming decades.

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